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Reuse of materials from a Sustainable Drainage System device: Health, Safety and Environment assessment for an end-of-life Pervious Pavement Structure

Keywords: Sustainable Drainage System; Pervious Pavement System; Occupational Health; Environmental Health; Gas Chromatography (GC) with Flame-Ionization Detection (GC/FID); inductively coupled plasma (ICP) spectroscopy; Sustainability, Waste Management

Abstract: Pervious pavement systems can have a life span of about 20 years but, at end-of-life, it becomes necessary to evaluate the state of the infrastructure to determine whether they pose a health and safety risk to workers during dismantling, and also determine potential reuse of the waste material generated. In this paper, we report of an investigation conducted to evaluate whether Pervious pavement systems are hazardous to human health at end-of-life and also to assess the mobility of the stormwater pollutants trapped in the system as a measure of their potential release to receiving systems such as water-bodies and groundwater systems. After decommissioning, the pervious pavement structure was sampled for analysis including Gas Chromatography, inductively coupled plasma spectroscopy and, leachate analysis. Results show that carcinogenic risks were significantly below the regulatory limit of 1 x 10-6 while, the hazard quotients and cumulative hazard indices were also below regulatory value of 1, based on United States Environmental Protection Agency standards. Furthermore, mean concentrations of benzene, toluene, ethylbenzene and xylene were significantly less than the UK soil guideline values. The results of the leachate analysis show that the metals of concern, Pb, Zn, Cr, Ni, Cd and Cu were all below the threshold for reuse applications such as irrigation purposes as they were all below the regulatory limits such as Food and Agriculture Organization and, United States Environmental Protection Agency standards. Finally, the evaluation of potential reuse and recycling purposes indicate that wastes generated from the dismantling of the PPS are within limits for recycling as aggregates for other civil engineering projects as per European Union standards. This has potential to enhance UK's drive to achieve the target of 70% level of construction & demolition waste recovery for reuse and recycling by the year 2020 as per European Union Water Framework Directive.

Introduction

Sustainable Drainage Systems (SuDS) mitigate the environmental risks of urban runoff and the damage done by associated contaminants. So-called hard SuDS, such as Pervious Pavement systems (PPS) are designed to infiltrate and attenuate stormwater runoff to underlying layers of their structure. PPS also provide a suitable platform for car parking, pedestrian and vehicular traffic. The treated stormwater in the PPS is then gradually released to receiving natural water systems or channelled to constructed receiving systems such as storage tanks. In addition to the infiltration and detention of runoff to prevent flooding, it is well known that PPS also treats stormwater for the improvement of discharged water quality in order to protect receiving natural water systems (Shuttleworth et al., 2017; Mbanaso et al., 2016; Pratt et al., 1999). This cleaning function is an issue that generates the potential for stormwater reuse. Although the entire system works together to detain and treat stormwater before controlled discharge downstream, the polypropylene geotextile component has been identified as site of dense pollutant retention and biodegradation of pollutants (Bond et al., 1999; Newman et al., 2006; Mbanaso et al., 2014). The treatment processes which occur within the PPS structure include filtration of contaminants, adsorption onto surfaces, biodegradation by microbes and sedimentation of particulates. Some of the pollutants include heavy metals, hydrocarbons, nutrients, emerging organic contaminants and priority substances, which are all of environmental concern.

The geotextile fibre provides a conducive environment for pollutant degrading microorganisms to thrive by using trapped pollutants to form biofilms. A biofilm is an aggregate of microbes in which the cells of microorganisms stick together to become ingrained within a self-produced matrix of extracellular polymeric substance (Mbanaso et al., 2013). Scanning electron microscope (SEM) analysis of the biofilm structure found on geotextiles fibre of a PPS has shown a diverse community of microbes consisting of bacteria, fungi, protists and metazoans (Coupe, 2004). The retention and removal of contaminants through natural processes protect receiving water bodies (Bond et al., 1999; Coupe, 2004). Water discharged from a PPS or stored in the system for reuse is a potential source of irrigation water and other reuse applications such as fountain refill and flushing of toilets etc. (Nnadi et al., 2009; Mbanaso et al., 2016). However, after several years in service, pollutants and toxins can become trapped in the PPS and accumulate (Charlesworth et al., 2017). Previous research (Pratt, 2001) has expressed concerns over the pollutants that accumulate in these SuDS devices as contaminants which are either non-biodegradable or could not be completely degraded before the structures are dismantled, could leach from the device and may reach receiving water bodies. Furthermore, the potential risk posed by these substances such as hydrocarbons and heavy metals may become compounded because contaminants which were previously thought to be non-hazardous may become hazardous due to physico-chemical and biological reactions prevalent in the PPS system over many years. Water quality standards have been included in UK regulations as a result of the Water Framework Directive (EC, 2000). In order to achieve the Environmental Quality Standards required, the potential of SuDS are being increasingly harnessed. As SuDS receive contamination typically associated with road traffic and urban environments, the long-term accumulation of contaminants could potentially classify material as hazardous, an important consideration when undertaking maintenance procedures for operatives handling it, and when decisions have to be made about its eventual disposal. There is potential for the accumulated material to become hazardous to both human and environmental health. This may put the health and safety of workers at risk, especially those who handle such materials during maintenance, processing prior to disposal and others at disposal sites. Workers' Rights include the right to work in conditions that do not pose a risk of serious harm as well as provision of effective training about workplace hazards and control measures. According to Hughes and Ferrett (2009) demolition works is one of the most hazardous construction activities which has been found to be responsible for more deaths and major injuries than other construction operations. The impact of demolition works is not restricted to the immediate environment or site but, also involves the surrounding areas such as adjacent structures, people or passers-by and the milieu. For instance, in terms of volume, it is estimated that C&D waste constitutes the largest waste stream in the EU as it represents approximately a third of all waste generated (EU, 2016). Thus, efficient management of C&D waste is required as it potentially have several benefits in terms of sustainable development and improved quality of life. It also has potential to increase demand for C&D recycled materials and a major boost for the construction and recycling industry.

However, one of the major hurdles to re-using and recycling C&D waste is the lack of confidence in the quality of C&D recycled materials (EU, 2016). There are also concerns about potential health risks associated with C&D waste. These issues limit and impede the utilization of C&D waste and the development of the recycling industry. It is therefore, necessary to determine whether it is sustainable to dispose of these SuDS devices after dismantling them via re-use, recycling or other sustainable initiatives. It may well mean that the cost of disposal may be mitigated by the reclassification of the waste. Thus, there is the need for screening waste materials to accurately determine toxicity and prediction of the environmental fate of contaminants as hazardous waste regulations have become more stringent. Although PPS manage the environmental risks of urban runoff and encourage environmental enhancement particularly in car park areas, literature on the long-term resilience to the pollutants they store is scarce as well as information on their effectiveness at end-of-life. In this context, end-of-life is defined as the phase when the PPS which was previously in service, was dismantled.

There are studies which have demonstrated that a PPS can function satisfactorily without any maintenance for over 10 years (Pratt, 2004; Schlüter and Jefferies 2001; Wild et al.,

2002) and over 20 years (Imbe et al., 2002). Table 1 shows recommendations by various stakeholders and practitioners for PPS replacement or reconstruction in part or whole.

Different monitoring initiatives have been carried out, focusing on different parameters and performance indicators. Although a few of the studies have evaluated the performance of the PPS system at end-of-life, such as the hydrological performance (Sañudo-Fontaneda et al., 2018); pollutants retention (Newman et al., 2011); pollutants biodegradation (Mbanaso et al., 2013), none has assessed the potential occupational health and safety impacts at end-of-life as well as the sustainability of the waste disposal after dismantling. There is currently no documented field study on the state of a PPS several years after field operation in terms of efficiency and hazardous nature of the pollutants. Although PPS are efficient at trapping pollutants, it is not known how long the accumulation of these pollutants will take to affect the treatment efficiency of PPS and therefore, one of the design life criteria. Even with degradable products, the input rate might exceed degradation rate. Furthermore, it is unknown whether these materials are capable of reaching receiving water bodies if the threshold of the PPS capacity to hold pollutants is exceeded and thus, deteriorate the water quality of local water courses. It is therefore necessary to assess them for bioavailability of toxic and hazardous pollutants.

Another aspect that was investigated was the determination of the leachability of the contaminants trapped in PPS in order to assess both hazards to workers by direct skin contact and potential release to receiving systems such as adjoining farmlands, gardens etc. and water bodies. From the point of view of skin contact it is hard to find suitable standards for comparison and thus an attempt is made here to use surrogate standards as conservative estimates of acceptable concentrations. This type of pollutants, known as nonpoint diffuse pollutants are often more difficult to trace, monitor and control than point source pollution (EPA Victoria, 2012) because they are problematic to measure and are highly variable due to different climatic conditions and rain patterns (Lewis, 1999). The UK Environment Agency (EA) noted that "a brief review of the sources and mechanisms of diffuse pollution, monitoring systems and current regulatory controls indicate significant weaknesses in our ability to manage this problem in the UK" (Geological Society, 2003). It is essential to note that diffuse pollution is not only an environmental issue but also a human health and economic problem as well. If they are taken up by plants via agriculture, they may end up in the food chain which could become an issue of health concern and associated economic implication, hence the need to carry out an evaluation. If farmers spray their farms during irrigation with irrigation water containing these leachates, the process could expose them to inhalation of such fumes which could be a health concern. Thus, a range of limits for use of irrigation water are presented, both for direct comparison of the effects on the food chain and also, on the basis that if such water is being used in agriculture, skin contact would be almost inevitable and thus, any reported health effects due to exposure would have resulted in lower limits.

Since PPS is a sustainable device, it was further assessed to establish the sustainability of the waste materials after dismantling. As the global population increases, it is expected that it will exert consequential pressure on housing, infrastructural and industrial development. These activities generate an enormous amount of construction and demolition (C&D) waste which will require a balanced management approach to ensure sustainability of natural resources. Globally, C&D waste account for a major portion of all solid waste produced (Poon, 2007) because the construction industry is a principal consumer of natural resources thus, many of the eco-friendly initiatives focus more on improving the efficiency of resource utilization. For instance, in 2014, about 850 million tons of C&D waste was generated per annum by EU Member countries (Staunton et al., 2015; Fischer and Werge, 2009) out of which, UK and France accounted for about 90 million tons and 349 million tons per annum respectively (Williams and Turner, 2011; ADEME, 2017). In Japan, Hong Kong, Australia and USA, the quantities generated were 77 million tons, 15.4 million tons, 20 million tons and 534 million tons per annum respectively while China generated 200 million ton per annum (UNEP, 2015; Poon, 2009; Reardon and Fewster, 2013; US EPA, 2016). But, with stricter environmental regulations, increasing scarcity of land required for landfill, in addition to increased landfill cost, reuse of C&D waste is gaining global acceptance with increasing demand for aggregates. According to Tam et al. (2018), the global production of aggregate reached 40 billion tons in 2014, up from 21 billion tonnes recorded in 2007. This demand depletes natural resources and presents an ecological challenge which can be more sustainably met by augmenting with recycled aggregates produced from C&D waste which could possibly be reused. In developed economies such as Germany, UK and the USA, legislations and regulations are driving a scenario where a huge market is created for building materials and associated products which are generated from C&D waste materials (Tam et al., 2018). The generation of C&D waste takes place during the main phases of the life cycle of the building which are construction, renovation and demolition but, the demolition phase generates the greatest percentage of waste and therefore, requires most attention (Da Rocha and Sattler, 2009). In the European Union, the Waste Framework Directive (WFD) 2008/98/EC is the driver for sustainable management of wastes. Part of the waste management strategy is the End-of-waste (EoW) criteria which specify when certain waste ceases to be waste and obtains a status of a product (or a secondary raw material). According to Article 6 (1) and (2) of the WFD, certain specified waste shall cease to be waste when it has undergone a recovery (including recycling) operation and complies with specific criteria to be developed in line with certain legal conditions, in particular:

- the substance or object is commonly used for specific purposes
- there is an existing market or demand for the substance or object;
- the use is lawful (substance or object fulfils the technical requirements for the specific purposes and meets the existing legislation and standards applicable to products);
- the use will not lead to overall adverse environmental or human health impacts (EC, 2016).

Since PPS is a SuDS device, it should be expected to be ecologically beneficial, sustainable and economically viable to establish the potential to convert the waste generated during the decommissioning phase of the PPS into useful products. It is therefore, necessary to assess them for conformity with EoW criteria. The main aims of this research are therefore threefold:

1. To investigate whether PPS is hazardous to human health at end-of-life.
2. To determine leachability of the contaminants and potential release to receiving systems and water bodies.
3. To determine whether the waste from PPS can be sustainably disposed of after dismantling.

This paper presents one of the series of on-going field studies trying to establish the actual state of a PPS when dismantled. Although this paper is directed at PPS, the trapping mechanism for pollutants apply to other SuDS e.g. filter strips, bioretention systems, filter drains etc. therefore, outcomes may be extended to these systems.

2.0 Methodology

The study described here was a field-based research site set on an 11-year old parking area built as an experimental test bed. It was constructed in Bury, Lancashire as a multi-bay block paved system, located at the site of a former 18th Century water-powered cotton mill with each single bay of dimensions 2400mm x 4800mm (Newman et al., 2014). The research site is as shown in Figure 1a while the schematic of the cross section of the test bed is as shown in Figure 1b. It was eventually dismantled after 11 years due to the relocation of the company and, the desire to get as much data as possible was the motivation for this study.

2.1. Sample collection

Throughout the working life of the test bed studied here, it would have been expected to have received a at least an average oil loading based on derived literature value of 9.27 g per m^2 $year^{-1}$ (Bond, 1999). However, since the parking area was in daily use at an industrial site and was regularly used for parking commercial vehicles and occasionally used for parking plant such as excavators, the loading may have been higher. The part of the test bed subjected to study here was an isolated 3-bay section constructed using Permavoid plastic void formers with a pervious concrete block wearing course and no artificial additions of hydrocarbons were made. This is in contrast to other parts of the test bed (all of which were hydraulically isolated) which were based on stone subbases of various types and alternative wearing courses. These sections had been dosed with artificially high loadings of lubricating oil and diesel (in one case, 5 litres of used lubricating oil and 15 litres of diesel per parking bay). These sections had been excavated and re-laid for experimental purposes some time before the work reported here. With the surface loading on these sections, there would be a possibility that there would be transfer on vehicle tyres from the dosed sections to the section studied here. Thus, the data reported in this paper is unlikely to have been derived from a parking area that had been unusually lightly contaminated through its life.

Some of the results obtained during monitoring regimes have already been published (Newman et al., 2014; Newman et al., 2015). Sample collection was usually carried out after natural rainfall but in the case of the high loading experiments, simulated rain events were also created using a hosepipe and sprinkler. The results indicated by these papers were that in general use, the release of pollutants from such parking systems is minimal even after ten years of operation and that even under high loading events by far the majority of applied hydrocarbons were retained in the system, for the particular pavement designs to which they were applied (see Newman et al., 2015).

2.2. Decommissioning

At the end of the lifespan of the test bed, it was dismantled to identify the location and concentrations of the pollutants. Individual pieces of the pavement structure, aggregates and loose sediments were then carefully removed using a spatula and stored in labelled plastic bags and refrigerated prior to shipment to the analytical laboratory. The sampling was carrying out in hierarchical order according to the structure of the PPS from the pavement blocks down through the various layers of the system i.e. pavement blocks (P), Aggregates Alone (AA), Aggregates and Dust (AD), Dust alone (D) and Geotextile fibre. Approximately 2.5 kg each of P, AA, AD, D and 250 g of geotextile fibre were sampled for analysis.

2.3. Analytical Methodology

Analysis was carried out by ALS Environmental Ltd laboratory which is accredited by the United Kingdom Accreditation Service (UKAS) to ISO/IEC 17025 standard, the Drinking Water Inspectorates' Drinking Water Testing Specification (DWTS) and the Environment Agency of England & Wales Monitoring Certification Scheme (MCERTS).

The determinands and the principle of methods are as follows:

2.3.1. Monoaromatic Hydrocarbons

Monoaromatic hydrocarbons including benzene, toluene, ethylbenzene and o-m-p-Xylenes (BTEX) were extracted and analysed. A known amount of soil, with a surrogate spike added, was shaken and sonicated in an extraction vial containing pentane and acetone. Water was added to the sample and then centrifuged, a test portion of the resulting pentane layer was transferred to a 2ml vial and internal standard added. The extract was then analysed using Gas Chromatography (GC) with Flame-Ionization Detection (GC/FID).

2.3.2. Heavy Metals

The test samples were prepared in accordance with method BSEN 12457.3. Prior to analysis, a portion of the sample was diluted 20-fold with a blank standard. The concentration of the metals was determined by inductively coupled plasma (ICP) spectroscopy.

2.3.3. Leachate Analysis

Soluble and suspended species are leached from waste into water. The procedure is based on BSEN 12457-3:2000 and consists of a two-stage batch test at a liquid to solid ratio of 2 litres per kg and then 8 litres per kg. The sample material, which originally or after pre-treatment, had a particle size below 4mm, was brought into contact with water under defined conditions. The solid residue was separated by filtration and the leachate tested for temperature, conductivity and pH to give an indication of leachate conditions.

Leachate tests give an indication of the potential of the substance to percolate after being mobilised. The release of pollutants from the solid mass could be as a result of the movement of a liquid passing through to the solid, thereby extracting soluble or suspended solids or mobilising the contaminants into the liquid. It is thought that the C&D waste could leach to adjoining farmlands, surface waters or other water bodies. Farmers who irrigate their farms with water containing these leachates may be exposed to health hazards, especially through inhalation during spraying. In the current context, values obtained were evaluated against irrigation water limits on the assumption that if it is permissible for irrigation, this must be a conservative limit for workers both in skin contact or droplet inhalation.

2.4. Human Health Risk Assessment

Risk assessment is a contemporary approach for the identification of hazards and risk factors that have the potential to cause harm to humans and the environment, so that effective strategies can be developed to eliminate or minimize impacts (Zhang et al., 2012). The concept of risk assessment is a globally-accepted method for the evaluation of potential health effects of carcinogens and non-carcinogens (Moolla et al., 2013; Durmusoglu et al., 2010). In order to assess the human health risks associated with exposure to toxic substances on site, carcinogenic risk assessment (CRA) and Hazard Quotient (HQ) were applied to determine potential risks of carcinogenic and non-carcinogenic hazards respectively (US EPA, 2010a, Masih et al., 2016). These calculations were based on inhalation route only as it is the principal route of exposure to humans (ATSDR, 2007) as existing evidence indicates that these toxic substances such as heavy metals, can travel long distances while airborne and readily absorbed by humans following inhalation (Dinis and Fiuza, 2011; Jan et al., 2015).

The CRA or Cancer risk was calculated using the following equation (Chen et al. 2012; US EPA 1989):

CRA = (I) x (CPF)………… (1)

where *I* is the average daily intake due to exposure to the substance measured in mg kg^{-1} day^{-1} while *CPF* is the carcinogen potency factor or slope factor which represents the slope of the dose-response curve also expressed in mg kg^{-1} day^{-1}. CRA is an expression of a possibility that an individual exposed to a carcinogenic substance may develop cancer over the period of lifetime exposure (Ferreira-Baptista and Miguel, 2005; Zheng et al., 2010). For regulatory purposes, the acceptable or tolerable risk level is in the range of $1 \times 10^{-6} – 1 \times 10^{-4}$ (US EPA, 2001a; US EPA, 1986). Thus, if CRA > 1×10^{-4}, it is considered to be harmful to humans, but if CRA < 1×10^{-6}, it is regarded as negligible and acceptable.

Non- carcinogenic risk is expressed in terms a HQ and the applicable equations to calculate HQs for Toluene, Ethyl Benzene and Xylenes (TEX) are as follows:

$$\mathbf{HQ} = \frac{\mathit{I}}{\mathit{RfD}} \quad \text{……………………………………} \quad \mathbf{(2)}$$

where *I* is the average daily intake and RfD is the reference dose expressed in mg kg^{-1} day^{-1} (US EPA, 2010b; US EPA, 1993). If HQ > 1, then there is an adverse health effect of concern but, when HQ ≤ 1, then it is within acceptable level as there is no possibility of any adverse health effect (US EPA, 1986).

The average daily intake, I, is computed by considering several factors such as inhalation rate, frequency of exposure, duration of exposure, body weight of those exposed etc. It is calculated by using the following equation:

$$\mathbf{I} = \frac{\mathbf{C * CF * IR * EF * ED}}{\mathbf{PEF * BW * AT}} \quad \text{……………………………………} \quad (3)$$

where C is the contaminant concentration (mg kg^{-1}); CF is the conversion factor (kg mg^{-1}); IR is the inhalation rate (m^3 day^{-1}); EF is the exposure frequency (days $year^{-1}$); ED is the exposure duration (years); BW is the average body weight (kg); AT is the average time (days). According to US EPA (1989), the standard average body weight for an adult is 70kg while daily inhalation rate is $12.8m^3$ day^{-1}. In the present study, the risk was calculated based on 8 working hours per day (except Sundays) and 30 days annual leave. The exposure frequency is thus calculated by using the following expression as documented (Moolla et al., 2013; Durmusoglu et al., 2010):

[52*(number of days worked/3)] – Number of days on annual leave ………………… (4)

[52*(6/3)] – 30] = 74

where the numbers are denoted as follows:

- 52 = number of weeks in a year
- 3 = 24 hours in a day/8 hours of work

The summary of the factors and their respective values are as shown in Table 2.

For human health risk assessment due to exposure to heavy metals present in the PPS, the assessment models represented by equations 1 – 4 were also adopted based on US EPA (2001b and 1986). Similarly, the CPF and the RfD of the metals are as shown in Table 3.

3. RESULTS & DISCUSSION

3.1. Monoaromatic Hydrocarbons (BTEX)

The analytical data from end-of-life analyses are as shown in Table 4. It was decided to profile the samples from top to the bottom of the PPS to have a view of contaminants distribution pattern as follows: Pavement blocks (P), Aggregates just below the blocks (AA), further down the profile for Aggregates and Dust (AD), dust just above the geotextile fibre (D) and the geotextile fibre (G).

From Table 4, results obtained were not precise and, were attributed to some analytical challenges due to matrix interference. Reporting Limits raised for VOC HS Soils due to sample matrix interference throughout. The interference may be due to the relatively high levels of hydrocarbons deposited in the system over the long period in service especially, on the geotextile fibre. Since the results were all "less than" values (i.e. <), assessment "as is" would be unrealistic. Hence, a worse-case scenario was assumed by equating all the "less than" values as the "actual values".

The heavy metals concentration is as presented in Table 5.

Matrix interference was also experienced in the analysis of heavy metals. Generally, the heavy metals displayed different patterns in terms of concentration levels found in different layers of the PPS. For Pb, the concentration decreased from the top pavement blocks but, started to increase from the Dust-only layer and was highest in the geotextile fibre. For Cr, it decreased from 0.057 mg kg^{-1} recorded for the pavement blocks and then remained constant throughout the remaining layers of the PPS at < 0.025 mg kg^{-1}. For Ni, it was the same concentration level throughout the entire layers of the PPS. Zinc increased from the top pavement blocks to aggregates-only and aggregates-plus-dust layers which were same, then decreased in dust-only layer before increasing to <0.25 mg kg^{-1} in the geotextile layer. Zn is an important element which has a diverse and very vital functions in humans such as, a requirement essential for growth and development (especially, the brain), accelerated healing of wounds, the body's defensive (immune) system, cell division and growth and breakdown of carbohydrates. Since it is an essential element due to nutritional or dietary requirement, the Permissible maximum tolerable daily intake (PMTDI) is a factor to be considered in analysing the toxic profile. Zn is a cofactor of the superoxide dismutase enzymes and also an important component of DNA which acts to stabilise phosphate groups and co-ordinate with organic bases (EMEA, 2007). The recommended limits for tolerable intake of Zn can be unclear, because there is a wide margin between nutritionally required amounts of zinc and toxic levels. For instance, WHO recommends a PMTDI of 0.3-1.0 mg kg^{-1}, which corresponds to 18-60 mg day^{-1} for a 60 kg adult (EMEA, 2007) whereas WHO recommended average dietary zinc intake in adults is 14-20 mg day^{-1} (WHO, 2018). At concentration levels recorded in this study, they are well within the limits required by for effective functioning of human body.

3.3. Leachate Analysis

The data from the Leachate analysis which was carried out on the PPS profile from top of the structure to the bottom made up of pavement blocks, aggregates alone, aggregates and dust, dust alone and finally, the polypropylene geotextile fibre are as shown in Table 6.

From Table 6, Zinc showed the same concentration level (< 0.025 mg kg^{-1}) evenly distributed throughout the sections of the PPS under L/W ratio of 8:1 but, showed a variation under L/W ratio of 2:1. Chromium recorded similar values (< 0.0025 mg kg^{-1}) under both types of L/W ratio in all layers of the PPS except in the pavement block layer where it was 0.0053 mg kg^{-1} and 0.0058 mg Kg^{-1} under L/W ratios of 2:1 and 8:1 respectively. Nickel recorded the same value (< 0.02 mg kg^{-1}) under both L/W ratios in all sections of the PPS. Cadmium was similar to Nickel in terms of having same value throughout while Copper recorded some variations under both L/W ratios. The origins of the metals in the materials produced from the dismantling of the pavement are largely but not exclusively derived from automobiles. Copper, for example is released from brake linings and zinc is released from tyre wear materials. A wide range of wear metals are released from engine exhausts. Whilst one may expect chromium to be included here, most chromium probably comes from the wear of the concrete blocks. Cement used to be associated with chromium VI derived dermatitis in building workers but nowadays, reducing agents are included in cement to keep the chromium in the trivalent form. It may be useful in future studies to determine the hexavalent as well as total chromium.

3.4 Sustainability of waste analysis

According to a Technical Report by the Joint Research Centre, the European Commission's in-house science service whose aim is to provide evidence-based scientific support to the European policy-making process, leaching tests should be used to evaluate conformity with EoW criteria (EU, 2014). The report stated that leaching tests should be used to evaluate potential release of substances from aggregates to soil, surface water and groundwater rather than analysis of total composition (content), because the leaching properties of an aggregate are directly related to risk of such impacts. The leaching limit values and results for utilisation of waste-derived aggregates in some EU Member States are as presented in Tables 7 and 8 respectively.

Pavement blocks (P)

From Table 8, data showed that <0.06 mg kg^{-1} of Antimony (Sb) was leached under the liquid to waste ratio of 10:1 from the pavement blocks whereas the inert waste limit for Sb is 0.06 mg kg^{-1}. Even if it is assumed that the highest possible value of 0.06 mg kg^{-1} was recorded, it is not higher than the leaching limit values and therefore, of no environmental or health concern. Furthermore, the contaminant transport modelling which is required to determine the impacts due to soil properties such as bulk density, adsorption, saturated hydraulic conductivity, dilution etc. have not been evaluated. Similar trends were also recorded for Selenium (Se) and Phenol index with concentrations of <0.10 mg kg^{-1} and <1.0 mg kg^{-1} as against inert waste limits of 0.1 mg kg^{-1} and 1 mg kg^{-1} respectively thus, both Se concentration and Phenol Index do not indicate any leaching potential from the pavement blocks.

Aggregates alone (AA)

Table 8 showed that the concentration of the TDS from the aggregates-only sample dropped considerably from 6467 mg kg^{-1} recorded for the pavement blocks to 1300 mg kg^{-1} for the aggregates only sample. This may be due to the nature of the aggregates used which did not seem to present an attractive site for dissolution of the solids. The concentration levels of

Sb, Se and the Phenol Index recorded for the pavement blocks were similar to those of the aggregates-only sample. These indicate that these pollutants were generally distributed within these sections of PPS without any observable preference.

Aggregates and dust (AD)

From Table 8, the concentration of Barium (Ba) decreased with decreasing depth as it decreased from 3.3 mg kg^{-1} found in the pavement block to <0.6 mg kg^{-1} found in both the aggregates-only and aggregates plus dust sections. This indicates that most of the Ba was trapped in the pavement blocks. Some of the health effects traced to exposure to Ba includes damage to lungs, hypokalaemia, which can result in ventricular tachycardia, hypotension and paralysis. However, this study indicates that they are all within the regulatory limits and therefore, not a source of environmental or health concern.

Dust alone (D)

Like the figures obtained in the pavement blocks, Table 8 showed that Dust-only profile shared some similarities in terms of concentrations of Sb, Se and Phenol Index. It was also <0.06 mg kg^{-1} of Antimony (Sb) and 0.1 mg kg^{-1} of Selenium (Se) while Phenol index was <1.0 mg kg^{-1}. In our current context, they are all within limits of criteria required to be considered as waste-derived aggregates with EoW status.

Polypropylene geotextile fibre (G)

From Table 8, it can be observed that all the metals are within the regulatory limits for consideration as recyclable aggregates based on the stipulated criteria.

4.0 Potential for Environmental and Health Impacts

4.1 Exposure limit evaluation of BTEX

The interferences present in the BTEX analysis pushed up the reporting limits that were presented by the laboratory. Thus, all the reported values for BTEX were less than figures. This presented some problems as to interpret the results. A very conservative approach was chosen in that the assumption was made that the actual measured concentration was at the less than figure reported. Since there was great uncertainty in the actual concentrations a direct risk assessment approach was discarded in favour of using secondary standards. The standards selected were the UK contaminated land soil guideline values adopted for commercial sites (CL:AIRE, 2018). These limits take into account both inhalation risk and skin contact (and allow for a limited amount of direct ingestion of soil) and take into account the risk to workers on the site who are considered to be the sensitive receptor. As they are generally aimed at protecting a worker over an entire working lifetime they were considered to be very conservative limits. Table 9 represent analytical values versus the acceptance criteria.

As mentioned previously in the current study, analytical issues made it impossible to obtain actual values due to matric interference but, even if the highest less-than values obtained were adopted, they were still far lower by several orders of magnitude than the UK CLEA model limits. This suggests no potential health risks associated with handling of these decommissioned products.

4.2 Heavy Metals

Occupational exposure to heavy metals in the workplace has the capacity to cause many human health effects, ranging from pulmonary and cardiovascular inflammation to tissue damage and cancer. Under the UK Control of Substances Hazardous to Health (COSHH) Regulations, control measures to effectively deal with exposure to these hazardous substances is said to be adequate only if the following three conditions are satisfied. Firstly, any Workplace Exposure Limits (WELs) are not exceeded; secondly, the principles of good control practice are applied; thirdly, in cases where residual risks still exist, exposure to asthmagens, carcinogens and mutagens are reduced as low as reasonably practicable (HSE, 2013). The essence of Occupational health is to consider the effect that work may have on health and the effect that health can have on work itself with the sole aim of preventing ill-health instead of curing it. Under the UK Health and Safety at Work Act 1974 (HASAWA), employers have a legal duty to reduce risks (so far as reasonably practicable), to the health and safety of employees and others who may be affected by their work activity. Generally, the starting context is to assess the risks and, if the risk assessment is sufficiently and efficiently carried out, it will highlight where there is a significant residual risk to health even after reasonably practicable control measures have been applied (HSE, 2007). If there is a significant residual health risk to employees then, Health Surveillance should be considered. Adequate training should be provided to employees to understand their role and responsibilities within a health surveillance program and attendance made mandatory especially, where a risk assessment has established that a genuine need for health surveillance exists. According to HSE (2005), Health Surveillance enables a procedure to be put in place for early detection of work-related ill health which will trigger early response based on results obtained. Thus, the establishment of Health Surveillance is not an end in itself but indicates efficiency and effectiveness of existing control measures in reducing and avoiding workplace health hazards (HSE, 2007). If existing controls are deficient, then, employers should then provide suitable and sufficient controls to demonstrate they are meeting their duty of care for their employees. In the current study, Table 10 represents the estimation of individual heavy metal pollutant cancer and non-cancer risks based on equations 1 – 4.

The CRA of some of the selected heavy metals, Pb, Cr, Ni ad Cd shows that the cumulative risk level throughout the PPS profile was of the order Cr > Ni > Cd > Pb with values of 6.0×10^{-18}, 9.0×10^{-19}, 2.9×10^{-20} and 4.3×10^{-20} respectively. These figures are significantly less than the regulatory limit of 1×10^{-4}; suggesting no health issues associated with heavy metals toxicity.

In terms of cumulative hazard index, Zn gave the highest cumulative non-cancer risk across the different PPS profiles followed by Cd, Cr, Pb, Cu and Nickel with values of 1.8×10^{-14}, 1.8×10^{-15}, 5.2×10^{-15}, 2.9×10^{-16} and 1.7×10^{-17} and 4.6×10^{-17} respectively. The cumulative non-cancer hazard indices (HI) calculated for all metals at PPS profiles P, AA, AD, D and G were $6.32 \text{ X } 10^{-15}$, 4.96×10^{-15}, 4.85×10^{-15}, 4.86×10^{-15} and 4.88×10^{-15} respectively. All these values are significantly below the regulatory threshold of 1 (i.e. HI < 1); indicating no potential adverse impacts associated with occupational exposure to these toxic metals during the decommissioning of PPS system.

4.3 Leachates

Leachate tests are essentially carried out to mimic how wastes will behave once buried in a landfill which also helps to facilitate understanding and determination of the potential to contaminate the environment especially, adjoining farmlands and water-bearing aquifers. The infiltrating rainwater provides a platform in which waste, particularly organics, can be broken down via degradation into simpler substances through a variety of biochemical reactions involving oxidation and reduction, hydrolysis, dissolution and microbial action (Taylor and Allen, 2006). Landfill leachate may occur due to water infiltration through the solid wastes, breakdown of the waste via biological degradation, suspension of particulate matter, solubilisation of soluble salts contained in the waste and corrosion of the wastes. Contamination of the environment especially, farmlands, surface waters, rivers and water-bearing aquifers is a major health and safety hazard and would require the waste producer to take appropriate action to establish suitable disposal route of minimal environmental impact. In circumstances where the leachates infiltrate into adjoining farmlands, inhalation via spraying of irrigation water containing these leachates by farmers may also have adverse effects if they are above the required regulatory limits. Hence, irrigation limits were used to assess whether leachate meets the standard for use as irrigation, noting that if it is suitable for irrigation where it posed no risk via inhalation, then, it will not pose any risk to construction workers.

In the current study, the results of the leachate analysis from different layers of the PPS (Table 6) show that all the metals of concern, Pb, Zn, Cr, Ni, Cd and Cu were all below the threshold for use for irrigation purposes. They were all below the derived effluent standard

(Newman et al., 2013), irrigation water limits (Rowe and Abdel Magid, 1995; Harivandi, 1982; Nnadi et al., 2014), FAO (1992) Wastewater quality guidelines for agricultural use, US EPA (2012) 2012 Guidelines for Water Reuse; Recommended water quality criteria for irrigation by a significant margin. The forgoing indicates that the effluent from this system may be used for irrigation purposes without any potential adverse health issues especially, from agricultural produce or inhalation by farmers via spraying to irrigate farm lands.

4.4 Sustainability of decommissioned PPS

Generation of solid waste is a common occurrence in both developed and developing economies. These solid wastes usually end up in landfill sites where they may cause soil, water and air pollution from their constituent materials. Due to depletion of land resources for landfilling, high landfilling costs and tougher environmental legislations, there is a global re-appraisal of waste management methods aimed at optimizing re-use and recycling of C&D waste. Some of the generated C&D waste can be re-used and recycled as aggregates for other civil engineering projects. As the global production of aggregates continues to increase (40 billion tons in 2014) (Tam et al., 2018), there is the need to augment this level of production to minimize impact on the natural environment and enhance sustainable use of resources. In the European Union, WFD is the main driver for assessment and reclassification of waste as a product subject to free trade and use (though only for specific purposes) (EU, 2014). The WFD currently has a set goal for member states to achieve a target of 70% level of C&D waste recovery for re-use and recycling by the year 2020. Recovered C&D waste can be re-used and recycled to provide substantial economic gain and significant environmental benefit. But, statistics show that there is a variation in the level of recovery for reuse and recycling among member states. For instance, the UK recycled 62% of total generated amount of C&D in 2011, Germany recycled 91% while France recycled 50% of C&D waste in 2014 (Tam et al., 2018). In the United States, it was 48% in 2011 (Randell et al., 2014) while Hong Kong recorded 38% recovery rate (Tam and Tam, 2008). In the current study, Tables 6 – 10 indicate that the waste generated from the dismantling of the PPS are within limits to be considered for recycling as aggregates. In the UK, PPS waste will require further processing because UK is the only EU member state which defines national EoW criteria for waste-derived aggregates with reference to WFD (EU, 2014). The Environment Agency (EA) takes decisions on a case-by-case basis as to whether waste should be considered as a resource based on Environment Protection (Duty of Care) Regulations (Bohmer et al., 2008). This further environmental impact assessment is carried out by the EA using leaching characterisation data as input to the assessment model. With the mandate to curtail unnecessary landfilling of materials and facilitate the use of waste as resource via the Waste Protocols Project, the EA and Waste and Resources Action Programme (WRAP) produced the Quality protocols to enable recovered products to be used without the need for waste regulation controls. This Quality protocol titled "End of waste criteria for the production of aggregates from inert waste", is facilitating the reduction of demand on primary aggregates, thereby helping the industry to become more sustainable. According to the EA "Aggregates from recovered inert waste produced in accordance with the WRAP - Quality Protocol for the production of aggregates from inert waste, are not likely to be waste" (MPA, 2011). The UK aggregates industry produces approximately 210 million tonnes per annum from over 1500 quarries but, with over 90 million tonnes of C&D waste produced each year, a recycling rate of 90% could represents a substantial contribution of 81 million tonnes of reusable aggregates (TCS, 2013). This will increase sustainable use of depleting resources and reduce the amount being sent to landfill. However, data from the current study suggests that the waste would conform to the requirements because they were significantly lower than the inert limits being used as the baseline for further assessment.

Conclusion

SuDS such as PPS will continue to play a vital role in sustainable development in both developed and developing economies worldwide. However, when these structures have reached their end-of-life, they will need to be decommissioned. Findings from current study show that carcinogenic risks from toxic contaminants trapped in the "bioreactor" were below the regulatory limit of 1 x 10^{-6} while, the hazard quotients and cumulative hazard indices were also below the regulatory value of 1. These indicate that there are no potential adverse health impacts associated with occupational exposure to toxic substances trapped in the PPS during the decommissioning phase of PPS system. The results of the leachate analysis from different layers of the PPS show that all the metals of concern, Pb, Zn, Cr, Ni, Cd and Cu were all below the threshold for use for irrigation purposes as they were all below the regulatory limits such as FAO Wastewater quality guidelines for agricultural use, US EPA 2012 Guidelines for Water Reuse etc. Furthermore, the evaluation of the potential use of the generated C&D waste for re-use and recycling purposes as aggregates for other civil engineering projects indicate that the waste generated from the dismantling of the PPS are within limits to be considered for recycling as aggregates. Recovery and recycling of waste from the PPS has the potential to enhance the UK's drive to achieve the target of 70 % level of C&D waste recovery for re-use and recycling by the year 2020 which was set for

EU member states via the WFD (EC, 2016). However, as this study is based on a single site, further studies would be required to provide more information on the impact and contributions of wide range of factors such as pollutant loadings, geographical profile, installation characteristics, differences in design, type of SuDS device, maintenance, etc. dependently or /and independently.

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References

ADEME (2017) National Recycling Report (2005-2014), Agence de l'Environment et de la Maitrise de l'Energie, France (online) available from https://www.ademe.fr/sites/default/files/assets/documents/national-recycling-report-2005-2014-201705_synthesis.pdf (12/03/2017)

Agency for Toxic Substances and Disease Registry (ATSDR) (2007) Toxicological Profile for Benzene, U.S. Department of Health and Human Services, Public Health Service, (online) available from https://www.atsdr.cdc.gov/toxprofiles/tp3.pdf (17/03/2018)

Ali, M.U., Liu, G., Yousaf, B., Abbas, Q., Ullah, H., Munir, M.A.M and Fu, B. (2017) Pollution characteristics and human health risks of potentially (eco)toxic elements (PTEs) in road dust from metropolitan area of Hefei, China. Chemosphere 181, Pages 111-121

Böhmer, S., Moser, G., Neubauer, C., Peltoniemi, M., Schachermayer, E., Tesar, M., Walter, B., Winter, B. (2008): AGGREGATES CASE STUDY, Final Report referring to contract n°

150787-2007 F1SC-AT "Aggregates case study – data gathering" (study commissioned by JRC-IPTS), Vienna.

Bond, P.C. (1999) Mineral Oil Biodegradation Within Permeable Pavements: Long-Term Observations. Unpublished PhD Thesis, Coventry University, Coventry, UK.

Bond, P.C., Pratt, C.J. and Newman, A.P. (1999) A review of stormwater quantity and quality performance of permeable pavements in the UK. Proc. of 8th International Conference on Urban Storm Drainage, Sydney, Australia. pp 248 – 255.

Buranatrevedh, S. (2013) Health Risk Assessment of General Populations Exposed to Metals from an Aluminum Production Plant in Thailand. J Community Med Health Educ 3: 262. doi:10.4172/2161-0711.1000262

CASQA (2003) Stormwater Best Management Practice Handbook: New Development and Redevelopment (online) available from https://www.casqa.org/sites/default/files/BMPHandbooks/BMP_NewDevRedev_Complete.pdf (20/04/2018)

Charlesworth, S.M., Beddow, J. and Nnadi, E.O. (2017) The Fate of Pollutants in Porous Asphalt Pavements, Laboratory Experiments to Investigate Their Potential to Impact Environmental Health. International Journal of Environmental Research and Public Health 14 (666) doi:10.3390/ijerph14060666.

Chen, X.; Lu, X.; Yang, G. (2012) Sources identification of heavy metals in urban topsoil from inside the Xi'an second ring road, NW China using multivariate statistical methods. Catena 2012, 98, 73–78.

CIRIA (2015) The SuDS Manual (C753) (online) available from https://www.ciria.org/Memberships/The_SuDS_Manual_C753_Chapters.aspx (20/04/2018)

CL:AIRE (2018) Soil Guideline Values (online) available from https://www.claire.co.uk/information-centre/water-and-land-library-wall/44-risk-assessment/178-soil-guideline-values?showall=1&limitstart (11/09/2017)

Coupe, S.J. (2004) Oil Biodegradation and Microbial Ecology within Permeable Pavements. Unpublished PhD thesis, Coventry University, Coventry, UK.

Da Rocha, C.G. and Sattler, M.A. (2009) A discussion on the reuse of building components in Brazil: an analysis of major social, economical and legal factors. Resources, Conservation and Recycling, 54 pp. 104-112

Dinis, M.de L. and Fiuza, A. (2011) exposure assessment to heavy metals in the environment: measures to eliminate or reduce the exposure to critical receptors. In: L.I. Simeonov et al. (eds.), Environmental Heavy Metal Pollution and Effects on Child Mental 27 Development: Risk Assessment and Prevention Strategies, DOI 10.1007/978-94-007-0253-0_2, © Springer Science+Business Media B.V.

Durmusoglu, E., Taspinar, F., Karademir, A. (2010) Health risk assessment of BTEX emissions in the landfill environment. Journal of Hazardous Materials 176, (1–3), 870-877

EC (2000) The EU Water Framework Directive (WFD) (online) available from http://ec.europa.eu/environment/water/water-framework/index_en.html (12/03/2018)

EC (2016) Waste Framework Directive; End-of-waste criteria (online) available from http://ec.europa.eu/environment/waste/framework/end_of_waste.htm (14/15/2018)

Environment Agency (2015) Delivering Benefits through evidence: Cost estimation for SUDS - summary of evidence http://evidence.environment-agency.gov.uk/FCERM/Libraries/FCERM_Project_Documents/SC080039_cost_SUDS.sflb.ashx (02/02/2018)

EPA Victoria (2012) Point and nonpoint sources of water pollution (online) available from http://www.epa.vic.gov.au/your-environment/water/protecting-victorias-waters/point-and-nonpoint-sources-of-water-pollution (11/05/2018)

ESAG (2009) Environmental Site Assessment Guideline; DB11/T656-2009; ESAG: Adelaide, Australia.

EU (2014) Study on methodological aspects regarding limit values for pollutants in aggregates in the context of the possible development of end-of-waste criteria under the EU Waste Framework Directive; Joint Research Centre Institute for Prospective Technological Studies (online) available from http://susproc.jrc.ec.europa.eu/activities/waste/documents/Aggregates%20leaching%20Main.pdf (12/05/2018)

EU (2016) EU Construction & Demolition Waste Management Protocol (online) available from file:///C:/Users/ab0312/AppData/Local/Temp/Protocol%20Ares(2016)5840668-101016.pdf (12/09/2018)

European Medicines Agency (EMEA) (2007) Guideline on the specification limits for residues of metal catalysts (online) available from http://www.ema.europa.eu/docs/en_GB/document_library/Scientific_guideline/2009/09/WC500003586.pdf (12/10/2017)

FAO (1992) Wastewater treatment and use in agriculture; FAO irrigation and drainage paper 47 (online) available from http://www.fao.org/docrep/T0551E/T0551E00.htm (02/05/2018)

Ferreira-Baptista, L.; De Miguel, E. (2005) Geochemistry and risk assessment of street dust in Luanda, Angola: A tropical urban environment. *Atmos. Environ. 39*, 4501–4512

Fischer, C. and Werge, M. (2009) EU as a Recycling Society. Present recycling levels of Municipal Waste and Construction & Demolition Waste in the EU (online) available from http://avfallnorge.web123.no/article_docs/EUgjenvinningsstudie.pdf (12/09/2018)

Harivandi, M.A. (1982) 'The Use of Effluent Water for Turfgrass Irrigation.'. Cooperative Extension: California Rurfgrass Culture 32. Nos. 3 & 4. (online) available from https://agops.ucr. edu/turf/publications/ctc/ctc32_34.pdf (12.09.14.)

HSE (2005) Health Suivellance (online) available from http://www.hse.gov.uk/health-surveillance/index.htm (08/11/2017)

HSE (2007) Risk Management (online) available from http://www.hse.gov.uk/risk/index.htm (24/10/2017)

Hughes, P and Ferrett, E. (2009) Introduction to Health and Safety at Work: The Handbook for the NEBOSH National General Certificate. USA: Routledge

Imbe M., Okui H., Hashimoto C. and Musiake K. (2002) Monitoring and analysis of implemented infiltration system over past 20 years. Proc. 9th Int. Conf. on Urban Drainage, Global Solutions for Urban Drainage, Eds. E W Strecker and W C Huber, Portland, Oregon, USA. ISBN 0 7844 0644 8 (40644-009-003.pdf).

INTERPAVE (2006) CONCRETE BLOCK PERMEABLE PAVEMENTS (online) available from http://www.paving.org.uk/commercial/documents/faqv1.pdf (20/04/2018)

INTERPAVE (2008) Understanding permeable paving: Guidance for Designers, Developers, Planners and Local Authorities (online) available from https://assets.marshalls.co.uk/dam-svc/assetstore/interpave--understanding-permeable-paving-6174.pdf (20/04/2018)

Jan, A.T., Azam, M., Siddiqui, K., Ali, A., Choi, I. and Haq, Q.M.R. (2015) Heavy Metals and Human Health: Mechanistic Insight into Toxicity and Counter Defense System of Antioxidants. International Journal of Molecular Sciences (online) available from https://www.ncbi.nlm.nih.gov/pmc/articles/PMC4691126/ (14/06/2018)

Kamunda, C., Mathuthu, M and Madhuku, M. (2016) Health Risk Assessment of Heavy Metals in Soils from Witwatersrand Gold Mining Basin, South Africa. Int J Environ Res Public Health. 13(7): 663.

Lewis, M A. (1999) NON-POINT SOURCE POLLUTION. Presented at Urban Stormwater County Task Force Meeting, Pensacola Junior College Media Center, Pensacola, FL, 9 November 1999.

Li, R.Z.; Zhou, A.J.; Tong, F.; Wu, Y.D.; Zhang, P.; Yu, J. (2011) Distribution of metals in urban dusts of Hefei and health risk assessment. Chin. J. Environ. Sci. 32, 2661–2668.

Liu, C., Lu, L., Huang, T., Huang, Y., Ding, L. and Zhao, W. (2016) The Distribution and Health Risk Assessment of Metals in Soils in the Vicinity of Industrial Sites in Dongguan, China. Int. J. Environ. Res. Public Health, 13, 832; doi:10.3390/ijerph13080832

Masih, A., Lall, A.S., Taneja, A. and Singhvi, R. (2016) Inhalation exposure and related health risks of BTEX in ambient air at different microenvironments of a terai zone in north India. Atmospheric Environment (147) Pages 55-66

Mbanaso, F.U., Coupe S.J, Charlesworth S.M, Nnadi E.O. (2013) Laboratory-based experiments to investigate the impact of glyphosate-containing herbicide on pollution attenuation and biodegradation in a model pervious paving system. Chemosphere 90, 737 – 746.

Mbanaso, F.U., Coupe, S.J., Charlesworth, S.M., Nnadi, E.O., Ifelebuegu, A.O. (2014) Potential microbial toxicity and non-target impact of different concentrations of glyphosate-containing herbicide (GCH) in a model pervious paving system. Chemosphere 100, 34–41.

Mbanaso, F.U., Nnadi, E.O., Coupe, S.J. and Charlesworth, S.M. (2016) Stormwater recycling in landscaped areas: Effect of Herbicide Application on Water Quality. Environmental Science and Pollution Research 23: 15970. DOI 10.1007/s11356-016-6729-7

Mineral Products Association (MPA) (2011) MPA Aggregates Information Sheet: Recycled Aggregates and the WRAP Quality Protocol (online) available from http://www.mineralproducts.org/documents/Information_Sheet_Recycled_Aggregates_WRAP_QP.pdf (21/05/2018)

Moolla, R., Valsamakis, S.K., Curtis, C.J., Piketh, S.J. (2013) Occupational health risk assessment of benzene and toluene at a landfill site in Johannesburg, South Africa. WIT Transactions on The Built Environment, Vol 134.

Newman, A.P., Coupe, S.J., Smith, H.G., Puehmeier, T., Bond, P. (2006) The Microbiology of Permeable Pavements. 8th International Conference on Concrete Block Paving. November 6–8, 2006 California USA.

Newman, A.P., Nnadi, E.O., Duckers, L.J. and A.J. Cobley, A.J. (2011) Further developments in self-fertilising geotextiles for use in pervious pavements. Water Sci. Technol., 64 (2011), pp. 1333-1339

Newman, A.P., Aitken, D. and Antizar-Ladislao, B. (2013) Stormwater quality performance of a macro-pervious pavement car park installation equipped with channel drain based oil and silt retention devices. Water Research 47 (20) 7327-7336, http://dx.doi.org/10.1016/j.watres.2013.05.061

Newman, A.P., Puehmeier, T., Shuttleworth, A. and Christopher J Pratt, C.J. (2014) Performance of an Enhanced Pervious Pavement System Loaded with Large Volumes of Hydrocarbons. Water Sci Technol. 2014;70(5):835-42. doi: 10.2166/wst.2014.301.

Newman, A. P., Nnadi, E.O. and Mbanaso, F.U. (2015) 'Evaluation of pervious and macro-pervious pavements as harvesting systems for localized landscape and horticultural irrigation' In: K. Karvazy and V.L. Webster (Eds). World Environmental and Water Resources Congress 2015, '2015 World Environmental and Water Resources Congress'. Held 17-21 May 2015 at Austin, Texas. American Society of Civil Engineers.

Nnadi, E.O., Duckers, L.J., Newman, A.P., Coupe, S.J., and Puehmeier, T. (2009) Irrigation using Permeable Paving as the Source of Water: Effects of Recycled Water on the Soil; Plant Growth and Development. Proceedings of the GeoAfrica 2009 Conference, Cape Town, South Africa.

Nnadi, E.O., Newman, A.P. and Coupe, S.J. (2014) Geotextile Incorporated Permeable Pavement System as Potential Source of Irrigation Water: Effects of Re-Used Water on the Soil, Plant Growth and Development. Clean–Soil, Air, Water 2014,42(2), 125–132

Poon, C.S. (2007) Reducing construction waste. Waste Manage 27(12):1715–6.

Poon, C.S. (2009) Sustainable management of construction waste for Hong Kong (online) available from https://www.epd.gov.hk/epd/msw_consult/file/MSW_ENG_ch1.pdf (12/09/2018)

Pratt, C.J., Newman, A.P. and Bond, P.C. (1999) Mineral Oil Bio-Degradation within a Permeable Pavement Long Term Observations. Water Sci. Technol. 39 (2) 103-109.

Pratt, C.J. (2001) Sustainable Urban Drainage – A review of published material on the performance of various SuDS devices prepared for the UK Environment Agency. Coventry University, UK.

Pratt, C. J. (2004) SUSTAINABLE DRAINAGE: A Review of Published Material on the Performance of Various SUDS Components prepared for the UK Environment Agency. Coventry University, UK (online) available from https://www.susdrain.org/files/resources/evidence/SuDS_lit_review_04.pdf (20/04/2018).

Randell, P., Pickin, J. and Grant, B. (2014) Waste generation and resource recovery in Australia- Reporting Period 2010/11, 2014.

Reardon, C. and Fewster, E. (2013) Materials: Waste Minimization (online) available from http://www.yourhome.gov.au/sites/prod.yourhome.gov.au/files/pdf/YOURHOME-Materials-WasteMinimisation.pdf (12/09/2018)

Rowe, D.R., Abel-Magid, I.M., (1995) Handbook of Wastewater Reclamation and Reuse. CRC Press Inc., New York

Sañudo-Fontaneda, L.A., Andres-Valeri, V, C., Costales-Campa, C., Cabezon-Jimenez, I. and Cadenas-Fernandez, F. (2018) The Long-Term Hydrological Performance of Permeable Pavement Systems in Northern Spain: An Approach to the "End-of-Life" Concept. *Water* 2018, *10*(4), 497; doi:10.3390/w10040497

Schlüter W. & Jefferies C. (2001) Monitoring the outflow from a Porous Car Park Proc. First National Conference on Sustainable Drainage Systems, Coventry June 2001.

Shabbaj, I.I., Alghamdi, M.A., Shamy, M., Hassan, S.K., Alsharif, M.M. and Khoder, M.I. (2018) Risk Assessment and Implication of Human Exposure to Road Dust Heavy Metals in Jeddah, Saudi Arabia. Int. J. Environ. Res. Public Health 2017, 15(1), 36; doi:10.3390/ijerph15010036.

Shuttleworth, A.B., Nnadi, E.O., Mbanaso, F.U. and Newman, A.P. (2017) Plant Growth Trials on Harvested Water and Waste Sludge from a Macro-Pervious Pavement System in Central Scotland. World Environmental and Water Resources Congress 2017: Water, Wastewater, and Stormwater; Urban Watershed Management; and Municipal Water Infrastructure - Selected Papers from the World Environmental and Water Resources Congress 2017. American Society of Civil Engineers (ASCE), 24-38

Staunton, J., Williams, C.D., Morrison, L., Henry, T., Fleming, G.T.A. and Gormally, M.J. (2015) Spatio-temporal distribution of construction and demolition (C&D) waste disposal on wetlands: a case study. Land Use Policy, 49 (2015), pp. 43-52, 10.1016/j.landusepol.2015.06.023

Tam, V.M. and Tam, C.M. (2008) Re-use of Construction and Demolition Waste in Housing Developments, 2008, 24

Tam, V.W.Y, Soomro, M. and Evangelista, A.C.J. (2018) A review of recycled aggregate in concrete applications (2000 – 2017). Constcruction and Building Materials 172 (2018) 272 - 292

The Concrete Society (TSC) (2013) Construction and demolition waste recycling may be the key to recovery (online) available from https://www.sheehancontractors.co.uk/includes/This%20is%20concrete%20article.pdf (21/05/2018)

UNEP (2015) Global Waste Management Outlook. International Environmental Technology Centre, Osaka, Japan. 10.1177/0734242X15616055

US EPA (1986) *Superfund Public Health Evaluation Manual*; EPA/540/1–86; U.S. Environmental Protection Agency: Washington, DC, USA.

US EPA (1989) Risk Assessment Guidance for Superfund, volume I: Human Health Evaluation Manual (Part A), EPA/540/1-89/002, U.S. Environmental Protection Agency (US EPA), Office of Solid Waste and Emergency Response, Washington, DC.

US EPA (1993) Reference Dose (RfD): Description and Use in Health Risk Assessments; Background Document 1 A; Integrated Risk Information System (IRIS): Washington, DC, USA.

US EPA (2001a) Risk Assessment Guidance for Superfund: Volume III—Part A, Process for Conducting Probabilistic Risk Assessment; EPA540-R-02-002; U.S. Environmental Protection Agency: Washington, DC, USA.

US EPA (2001b) Risk Assessment Guidance for Superfund: Volume I Human Health Evaluation Manual (Part D, Standardized Planning, Reporting, and Review of Superfund Risk Assessments) Final Office of Emergency and Remedial Response U.S. Environmental Protection Agency Washington, DC 20460 (online) available from https://www.epa.gov/sites/production/files/2018-03/documents/175137.pdf (24/05/2018)

US EPA (2002) Child-Specific Exposure Factors Handbook; EPA-600-P-00e002B; National Center for Environmental Assessment: Washington, DC, USA.

US EPA (2010a) Estimation of Relative Bioavailablity of Lead in Soil and Soil-like Materials Using In Vivo and In Vitro Methods; OSWER 9285.7-77; Office of Solid Waste and Emergency Response, U.S. Environmental Protection Agency: Washington, DC, USA.

US EPA (2010b) Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (Part F, Supplemental Guidance for Inhalation Risk Assessment, EPA-540-r-070–002), Office of Superfund Remediation and Technology Innovation Environmental Protection Agency, Washington, D.C (2010)

US EPA (2012) 2012 Guidelines for Water Reuse; EPA/600/R-12/618 (online) available from https://nepis.epa.gov/Adobe/PDF/P100FS7K.pdf (21/04/2018)

US EPA (2016) Advancing sustainable materials management: 2014 fact sheet, United States Environmental Protection Agency, Office of Land and Emergency Management, Washington, DC 20460, (online) available from https://www.epa.gov/sites/production/files/2016-11/documents/2014_smmfactsheet_508.pdf (12/09/2018)

WHO (2018) Evaluations of the Joint FAO/WHO Expert Committee on Food Additives (JECFA) (online) available from http://apps.who.int/food-additives-contaminants-jecfa-database/chemical.aspx?chemID=4197 (16/10/2017)

Wild, T.C., Jefferies, C., and D'Arcy, B.J. (2002) SUDS in Scotland – the Scottish SUDS database Report No SR (02)09 Scotland and Northern Ireland Forum for Environmental Research, Edinburgh.

Williams, I.D. and Turner, D.A. (2011) Waste management practices in the small-scale construction industry. Thirteen Int Waste Manag Landfill Symp (online) available from https://eprints.soton.ac.uk/346322/1/003p_Williams.pdf (12/09/2018)

Zhang, Y., Mu, Y., Liu, V. Mellouki, A. (2012) Levels, sources and health risks of carbonyls and BTEX in the ambient air of Beijing, China. J. Environ. Sci., 24 (1) 124-130

Zheng, N.; Liu, J.; Wang, Q.; Liang, Z. (2010) Health risk assessment of heavy metal exposure to street dust in the zinc smelting district, Northeast of China. *Sci. Total Environ. 408*, 726–733

Figure 1a: Photo of the research site
b: Schematic cross section of the test bed made up of the pavement blocks (P), bedding layer comprising of aggregates, with the top section just below the pavement block sampled as aggregates alone (AA), middle section as aggregates and dust (AD) and the lower section as Dust alone (D). The drains are covered with a polypropylene geotextile fibre (G).

Table 1: Recommendations for replacement or refurbishment of PPS.

Reference	Recommended replacement/ refurbishment time (years)	Comments
CIRIA 2015	10 – 15	Replacement or reconstruction of part/whole PPS
INTERPAVE 2008; 2006	20	Without maintenance
California Stormwater Quality Association, CASQA 2003	15 – 20	Reconstruction and replacement
Environment Agency 2015	20 – 25	replacement or reconstruction of part of the whole PPS

Table 2: The summary of the factors and their respective values

Variable	Unit	Description	Value	Refs
C	mg kg^{-1}	Pollutant concentration in dust		Present study; Shabbaj et al., 2018
CF	kg mg^{-1}	Conversion Factor	1×10^{-6}	Li et al., 2011; Shabbaj et al., 2018
IR	m^3 day^{-1}	Inhalation rate of dust	12.8	US EPA, 2002; Li et al., 2011; Shabbaj et al., 2018
EF	days $year^{-1}$	Exposure frequency	74	Durmusoglu et al., 2010; Moolla et al., 2013
ED	years	Exposure duration	24	US EPA, 2002; US EPA, 2001b; Shabbaj et al., 2018
PEF	m^3 kg^{-1}	Particulate emission factor	1.36×10^{9}	US EPA, 2001b; Shabbaj et al., 2018
BW	kg	Average body weight	70	US EPA, 1989; ESAG, 2009
AT	days	Average time	25500	Durmusoglu et al., 2010

Table 3: Reference doses (RfD) in mg kg^{-1} day^{-1} and Carcinogen Potency Factors (CSF) for the different heavy metals.

Heavy metal	*carcinogen potency factor (CPF)*	Reference dose (RfD)
Pb	4.2×10^{-2} (d)	3.5×10^{-3} (a)
Zn	n/a (b)	6.5×10^{-5} (c)
Cr	4.1×10^{1} (d)	2.8×10^{-5} (a)
Ni	1.0×10^{0} (b)	2.0×10^{-2} (a)
Cd	6.3×10^{0} (d)	2.5×10^{-6} (b)
Cu	n/a (b)	4.0×10^{-2} (a)

[a] Ali et al. (2017)
[b] Liu et al. (2016)
[c] Buranatrevedh (2013)
[d] Kamunda et al. (2016)

Table 4: Mean concentration of monoaromatic hydrocarbons (BTEX) obtained from analysis where PPS profiles P, AA, AD, D and G represent Pavement blocks, Aggregates alone, Aggregates & dust, Dust alone, and Geotextile fibre respectively.

Pollutant/Units	**End-of-life analysis** Concentration (mg kg^{-1})				
	P	AA	AD	D	G
Benzene	< 0.10	< 0.10	< 2.5	< 2.5	< 15
Toluene	< 0.10	< 0.10	< 2.5	< 2.5	< 15
Ethyl Benzene	< 0.10	< 0.10	< 2.5	< 2.5	< 15
Total Xylenes	< 0.20	< 0.20	< 5.0	< 5.0	< 15

Table 5. Mean heavy metal concentrations where PPS profiles P, AA, AD, D and G represent Pavement blocks, Aggregates alone, Aggregates & dust, Dust alone, and Geotextile fibre respectively.

Metal	**End-of-life analysis** Concentration (mg kg^{-1})				
	P	AA	AD	D	G
Pb	0.125	0.09	0.073	0.12	0.13
Zn	0.179	<0.25	<0.25	0.143	<0.25
Cr	0.057	<0.025	<0.025	<0.025	<0.025
Ni	<0.20	<0.20	<0.20	<0.20	<0.20
Cd	<0.001	<0.0001	<0.0007	0.0006	<0.001
Cu	<0.10	<0.10	0.045	0.052	0.17

Table 6: Mean concentration of leachates obtained from the leachate analysis where PPS profiles P, AA, AD, D and G represent Pavement blocks, Aggregates alone, Aggregates & dust, Dust alone, and Geotextile fibre respectively.

Eluate Analysis		**Amount Leached**		**Derived Effluent Standard***	**Miscellaneous Irrigation Water Limit**		**FAO**/US EPA*** irrigation limits**
Liquid : Waste Ratio		**2:1**	**8:1**				
Mass of Raw Test Portion (MW) kg		0.179					
Mass of Dried Test Portion (MD) kg		0.175					
Moisture Content Ratio (MC) %		2.83					
Dry Matter Content Ratio (DR) %		97.24					
					Long Term	Short Term	
Metal	**PPS Profile**	**mg/L**	**mg/L**	**mg/L**	**mg/L**	**mg/L**	**mg/L**
Lead as Pb	P	0.032	0.013	0.144	5[a]	10[a]	5
	AA	0.025	< 0.01				
	AD	0.011	< 0.01				
	D	0.0165	0.011				
	G	0.015	0.013				
Zinc as Zn	P	0.026	< 0.025	1	5[b]	---	2
	AA	< 0.025	< 0.025				
	AD	< 0.025	< 0.025				
	D	0.0255	< 0.025				
	G	< 0.025	< 0.025				
Chromium as Cr	P	0.0053	0.0058	0.068	0.1[c]	---	0.1
	AA	< 0.0025	< 0.0025				
	AD	< 0.0025	< 0.0025				
	D	< 0.0025	< 0.0025				
	G	< 0.0025	< 0.0025				
Nickel as Ni	P	< 0.02	< 0.02				

	AA	< 0.02	< 0.02	0.4	0.5[b]	0.05[b]	0.2
	AD	< 0.02	< 0.02				
	D	< 0.02	< 0.02				
	G	< 0.02	< 0.02				
Cadmium as Cd	P	< 0.0001	< 0.0001	0.0018	0.01[a]	0.05[a]	0.01
	AA	< 0.0001	< 0.0001				
	AD	< 0.0001	< 0.0001				
	D	0.00013	< 0.0001				
	G	< 0.0001	< 0.0001				
Copper as Cu	P	< 0.01	< 0.01	0.2	0.2[a]	5[a]	0.2
	AA	< 0.01	< 0.01				
	AD	0.018	< 0.01				
	D	0.0375	< 0.01				
	G	0.021	0.017				

*See Newman et al. (2013) for the method by which these standards were derived- they were based on discharge to a stream in Scotland using a dilution rate of 20:1

**FAO (1992) Wastewater quality guidelines for agricultural use

***US EPA (2012) 2012 Guidelines for Water Reuse; Recommended water quality criteria for irrigation

[a] Rowe and AbdelMagid (1995)

[b] Harivandi (1982)

[c] Nnadi et al. (2014)

Table 7: Leaching limit values for utilisation of waste-derived aggregates in some EU Member States at Liquid: Waste Ratio (L/S = 10 l kg^{-1})

Eluate Analysis	Leaching limit values*			
	EU LFD** Inert	Netherlands	Germany	France
	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}
Arsenic as As	*0.5*	0.9	0.2	0.5
Barium as Ba	*20*	22	---	20
Cadmium as Cd	*0.04*	0.04	0.03	0.04
Chromium as Cr	*0.5*	0.63	0.25	0.5
Copper as Cu	*2*	0.9	0.6	2
Mercury as Hg	*0.01*	0.02	0.01	0.01
Molybdenum as Mo	*0.5*	1	---	0.5
Nickel as Ni	*0.4*	0.44	0.2	0.4
Lead as Pb	*0.5*	2.3	0.8	0.5
Antimony as Sb	*0.06*	0.16	---	0.06
Selenium as Se	*0.1*	0.15	---	0.1
Zinc as Zn	*4*	4.5	2	4
Chloride as Cl	*800*	616	500	800
Fluoride as F	*10*	55	---	10
Sulphate as SO4	*1000*	1730	500	1000
Total Dissolved Solids (TDS)	*4000*	---	---	4000
Phenol Index	*1*	---	0.4	---
Dissolved Organic Carbon (DOC)	*500*	---	---	---

*EU (2014) Technical Report by the Joint Research Centre, the European Commission's in-house science service

**EU Landfill Directive Inert limit (1999/31/EC) (EU, 2014)

Table 8: Leaching results from eluate analysis of the PPS component parts for utilisation of waste-derived aggregates at Liquid: Waste Ratio (L/S = 10 l kg^{-1})

Eluate Analysis	Leaching values				
	Pavement blocks (P)	Aggregates alone (AA)	Aggregates and dust (AD)	Dust alone (D)	Polypropylene geotextile fibre (G)
	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}	mg kg^{-1}
Arsenic as As	<0.050	<0.050	<0.050	<0.050	<0.050
Barium as Ba	3.3	<0.60	<0.60	<0.60	<0.60
Cadmium as Cd	<0.0010	0.00014	0.00072	0.00061	<0.0010
Chromium as Cr	0.057	<0.025	<0.025	<0.025	<0.025
Copper as Cu	<0.10	<0.10	0.045	0.052	0.17
Mercury as Hg	0.0034	0.00236	0.0045	<0.0050	0.008
Molybdenum as Mo	0.051	0.12	0.143	0.085	0.031
Nickel as Ni	<0.20	<0.20	<0.20	<0.20	<0.20
Lead as Pb	0.125	0.09	0.073	0.12	0.13
Antimony as Sb	<0.060	<0.060	<0.060	<0.060	<0.060
Selenium as Se	<0.10	<0.10	<0.10	<0.10	<0.10
Zinc as Zn	0.179	<0.25	<0.25	0.143	<0.25
Chloride as Cl	223	9.17	27.9	21.65	<37
Fluoride as F	1.6	1.03	1.37	1.5	0.93
Sulphate as SO4	50.66	586	893	605	190
Total Dissolved Solids (TDS)	6467	1300	1933	1400	860
Phenol Index	<1.0	<1.0	<1.0	<1.0	<1.0
Dissolved Organic Carbon (DOC)	43.33	8.93	20.67	31.5	54

Table 9: Mean concentration of BTEX values versus acceptance criteria.

Pollutant	End-of-life analysis Concentration (mg kg^{-1})					Acceptance criteria UK SGV* (mg kg^{-1} DW)
	P	AA	AD	D	G	
Benzene	< 0.10	< 0.10	< 2.5	< 2.5	< 15	95
Toluene	< 0.10	< 0.10	< 2.5	< 2.5	< 15	4400
Ethyl Benzene	< 0.10	< 0.10	< 2.5	< 2.5	< 15	2800
Total Xylenes	< 0.20	< 0.20	< 5.0	< 5.0	< 15	9300

UK SGV = UK Soil guideline values; DW = dry weight (CL:AIRE 2018)

Table 10: Estimation of heavy metal pollutant cancer and non-cancer risks

Pollutant	PPS profile	carcinogenic risk assessment (CRA)	Hazard Quotient (HQ)
Pb	P	5.8×10^{-21}	4.0×10^{-17}
	AA	2.1×10^{-20}	1.4×10^{-16}
	AD	3.9×10^{-21}	2.6×10^{-17}
	D	4.7×10^{-21}	3.2×10^{-17}
	G	7.8×10^{-21}	5.3×10^{-17}
Zn	P	---	4.0×10^{-15}
	AA	---	3.6×10^{-15}
	AD	---	3.6×10^{-15}
	D	---	3.6×10^{-15}
	G	---	3.6×10^{-15}
Cr	P	2.2×10^{-18}	1.9×10^{-15}
	AA	9.6×10^{-19}	8.4×10^{-16}
	AD	9.6×10^{-19}	8.4×10^{-16}
	D	9.6×10^{-19}	8.4×10^{-16}
	G	9.6×10^{-19}	8.4×10^{-16}
Ni	P	1.8×10^{-19}	9.3×10^{-18}
	AA	1.8×10^{-19}	9.3×10^{-18}
	AD	1.8×10^{-19}	9.3×10^{-18}
	D	1.8×10^{-19}	9.3×10^{-18}
	G	1.8×10^{-19}	9.3×10^{-18}
Cd	P	5.8×10^{-21}	3.7×10^{-16}
	AA	5.8×10^{-21}	3.7×10^{-16}
	AD	5.8×10^{-21}	3.7×10^{-16}
	D	5.8×10^{-21}	3.7×10^{-16}
	G	5.8×10^{-21}	3.7×10^{-16}
Cu	P	---	2.3×10^{-18}
	AA	---	2.3×10^{-18}
	AD	---	4.2×10^{-18}
	D	---	5.2×10^{-18}
	G	---	3.9×10^{-18}

Highlights

No potential adverse health impacts associated with occupational exposure to benzene, toluene, ethylbenzene and xylene

No potential health effects from exposure to leachates which may be reused

Carcinogenic and non-carcinogenic risks significantly below the regulatory limits

PPS demolition wastes may be re-used and recycled as aggregates

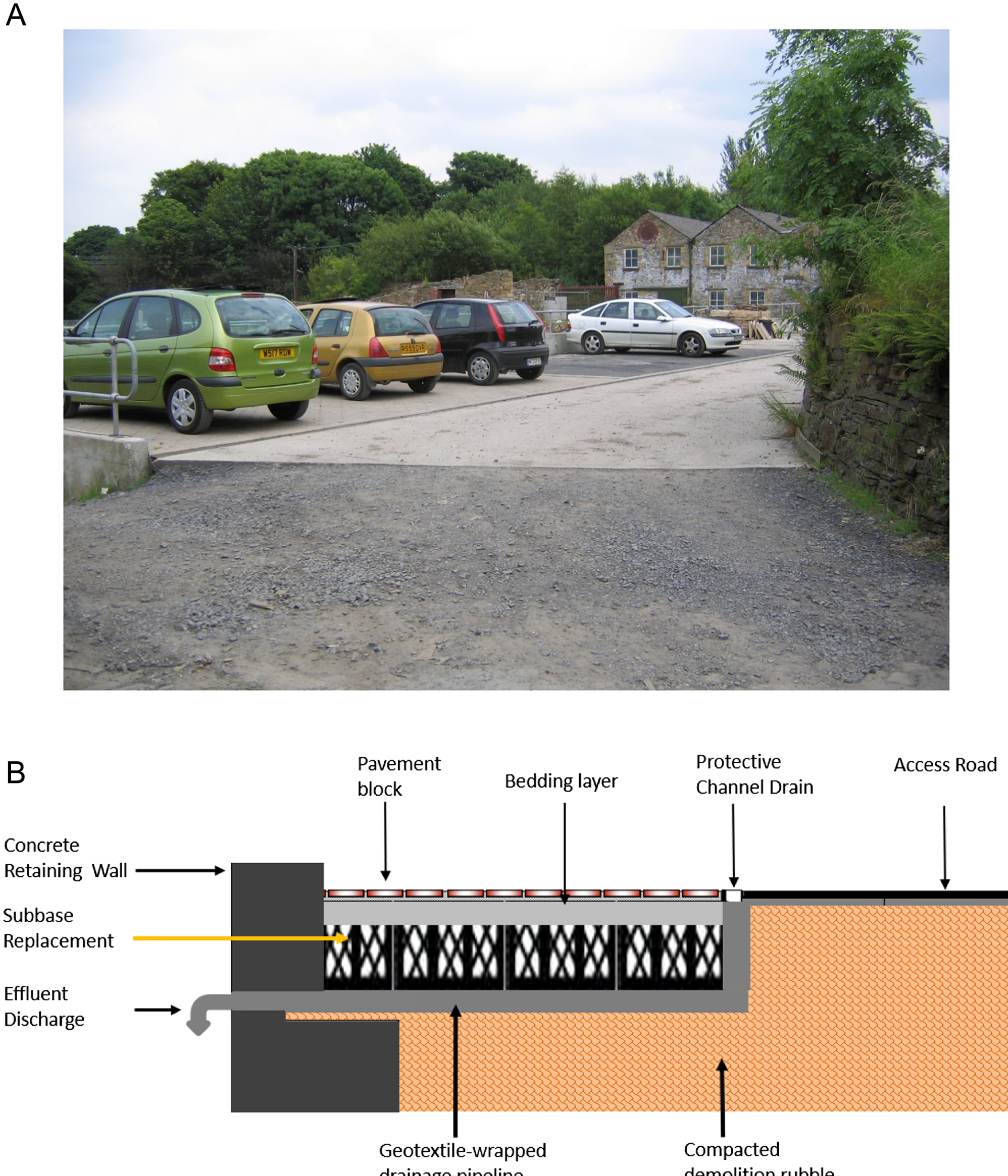
A
B
Pavement block
Bedding layer
Protective Channel Drain
Access Road
Concrete Retaining Wall
Subbase Replacement
Effluent Discharge
Geotextile-wrapped drainage pipeline
Compacted demolition rubble

Figure 1